Land Use in LCA (Subject Editor: Llorenç Milà i Canals)

Assessment of Land Use Impact on Biodiversity

Proposal of a new methodology exemplified with forestry operations in Norway

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Abstract

Goal, Scope and Background. Land use and changes in land use have a significant impact on biodiversity. Still, there is no agreed upon methodology for how this impact should be assessed and included in LCA. This paper presents a methodology for including land use impact on biodiversity in Life Cycle Impact Assessment and provides a case example from forestry operations in Norway.

Materials and Methods. The methodology presented applies indirect assessments of biodiversity based on knowledge on what key factors are important for maintaining biodiversity in a boreal forest. These are used to construct an index on Conditions for Maintained Biodiversity. In addition the intrinsic quality of an area is assessed on the basis of the Ecosystem Scarcity and Ecosystem Vulnerability. Globally available data on ecoregions are here used. In addition the spatial and temporal impact is assessed based on the annual regrowth of the forest.

Results. In the case study different forestry management regimes for the ecoregions 'Scandinavian and Russian taiga' and Scandinavian coastal coniferous forests' are compared. Based on the proposed methodology, the intrinsic quality difference of the two ecoregions is estimated to approximately 40% and the reduction in impact on biodiversity from land use by adopting new and realistic targets for the key factor 'areas set aside' is estimated to 20%.

Discussion. The paper presents a new methodology for how land use impacts on biodiversity can be included in LCA. The methodology is based upon a proposed framework and the results from the case study show that the methodology is capable to distinguish between different forestry management regimes and forestry in different ecoregions. The data used are readily available, but more research is needed to scale the proposed key factors and also include new key factors. It is at present not possible to validate the size of the differences.

Conclusions. The importance of land use impact on biodiversity is of major importance and should be included in LCIA. The proposed methodology is developed within a framework developed within the UNEP-SETAC Life Cycle Initiative and provides a methodology demonstrated to be able to distinguish between both similar activities in different ecoregions and different management practices within one ecoregion.

Recommendations and Perspectives. More work is needed to establish a methodology for land use impact on biodiversity in LCIA and due to the importance this should be a prioritized task. The proposed application of indirect indicators to assess impact on biodiversity from land use changes in LCIA should be further explored, but the proposed methodology can already be applied with globally available data on ecoregions. The challenge is to develop sound key factors for the relevant ecosystems.

Keywords: Biodiversity; ecoregion; forestry; key factors; land use impacts; land quality; LCA; LCIA

Introduction

According to Diaz and Cabido (2001) there seems to be no doubt that loss of biodiversity is one of the largest environmental problems, if not the largest. The main reason given for loss of biodiversity is changes in land use and a consequential unavoidable loss of habitats (Pimm et al. 1995, Chapin et al. 1998, Müller-Wenk 1998, Chapin et al. 2000, Sala et al. 2000). Still, there is no agreed upon method on how loss of biodiversity due to land use is to be included in life cycle assessments (LCA) (Milà i Canals et al. 2007) and it is even debated if this should be done at all (Udo de Haes 2006).

The land use impact on biodiversity is in particular important when extraction of raw materials originating from land extensive activities is assessed. Forestry as the origin for wood based products is a striking example (cf. Schweinle 2002). In Europe, the forested areas have increased with more than 9 million hectares during the last decade (UNEP 2002), but most of the natural forest vegetation is transferred to agricultural and urban areas, and most of what is left is strongly influenced by forestry and other human activities (Angelstam 1998, Larsson 2001).

Maintenance of biodiversity is an urgent issue for forestry operations (Angelstam 1998). In Norway, almost half of the species in the Norwegian Red List are forest living species (Kålås et al. 2006) and only 2.9% of the forested areas in the country can be classified as undisturbed by man (Hytteborn et al. 2005). Even though only a few species are known to be extinct, present forestry practice has given an extinction debt, i.e. species that are still present but are likely to go extinct in a not too far future due to present pressure (Angelstam 2001, Hanski and Walsh 2004). The extinction debt is not estimated in Norway, but is assumed to be approximately 1,000 of 20,000 forest living species in Finland (Hanski and Walsh 2004). As many as 50% of all species in Norway depending on dead wood are threatened (Framstad et al. 2002).

Loss of biodiversity is probably the major single environmental problem caused by the forestry sector (Seppälä et al. 1998). This aspect should thus be included in LCA on forest products to obtain a more holistic picture of the environmental impact of such products and enable comparison to other products (Schweinle 2002, Lippke et al. 2005, Milà i Canals et al. 2006). This would be in accordance with ISO 14040: 2006 that states that all aspects of natural environment must be considered. Land use is explicitly mentioned in ISO 14044: 2006.

In this paper the first outline of a methodology for assessing biodiversity aspects related to land use in forestry operations in a boreal forest is presented. One important characteristic with the proposal is the ability to distinguish both between different forestry management regimes and forestry at different locations. It is believed that the presented methodology could be applicable to other ecosystems as well, but this needs to be tested. This paper is also a contribution to the debate on how indicators for assessing land use impact on biodiversity should be framed (cf. Milà i Canals et al. 2006). The first attempt to apply the methodology on a case study of logging of spruce (*Picea abies*) in Norway is presented.

1 Background: Assessments of Biodiversity Applicable for LCA

Biodiversity is a concept with a wide content, and in the Convention on Biological Diversity it is stated that '*Biological diversity means the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems*' (UNEP 1992). In spite of this, the most frequently used indicator on biodiversity is number of species. Gaston (1996) claims there are four obvious reasons. First, species richness is thought by many to capture much of the essence of biodiversity, and many authors use the two terms more or less as synonyms. Second, species richness as term is widely understood. Third, species richness is considered in practice to be a measurable parameter in contrast to biodiversity as stated in the definition, and fourth, much data on species richness already do exist.

It is suggested that the low focus on conservation of biodiversity in decision making is due to the fact that biodiversity is hard to quantify (OECD 2002). The low focus on loss of biodiversity as a consequence of land use in LCA is most likely a result of this. Nevertheless, several attempts have been made to include land use in LCA (see Milà i Canals et al. 2007 for references), but proposed indicators are in most cases not checked with a consistent framework (Milà i Canals et al. 2007).

Some of the proposed methodologies, such as the Biotope Method (Kyläkorpi et al. 2005), are at present too coarse to distinguish between different management regimes. Another severe problem with many of the methodologies proposed is that they are based on assumptions that probably are invalid. This relates in particular to the proposal of vascular plant diversity as an indicator for biodiversity in some of the methods, e.g. the SPEP-method (Köllner 2000) incorporated in Eco-indicator 99 (Goedkoop and Spriensma 2001). The SPEP-method is a praiseworthy proposal, but might turn out as a dead end due to the underlying assumptions.

The first problem is that vascular plant diversity is an inappropriate indicator for biodiversity. An overwhelming number of studies show no correlation between species richness in one taxonomic group and species richness in other groups (i.e. Prendergast et al. 1993, Hengeveld et al. 1995, Gaston 1996, Dobson et al. 1997, Lawton et al. 1998, Molau and Alatalo 1998, Chapin et al. 2000, Larsson 2001). Lawton et al. (1998) conclude that on average only 10–11 percent of the variation in species richness of one group can be predicted by the change in richness of another group.

Also, if ecological changes are to be assessed through registration of changes in species composition, other taxonomic groups are more useful. Just to mention a few; Molau and Alatalo (1998) have shown that bryophytes are better indicators than vascular plants for effects of global warming, Hilmo and Holien (2002) have shown that lichens are useful indicators for edge effects and fragmentation, and Bongers (1990) has shown that nematodes are useful indicators for changes in soil conditions.

A third problem is that it is not only important what species that are present, it is also important to maintain areas that enable invasions. A focus on presence or absence of different species is more or less consciously based on an assumption of static conditions in the ecosystems. This is simply not true, cf. the equilibrium theory of island biogeography (MacArthur and Wilson 1967), the metapopulation concept (e.g. Schemske et al. 1994), and the natural disturbance hypothesis (Connell 1978). In addition, there might be a tremendous time lag between the changes in conditions[1] and the actual changes in species composition. Saunders et al. (1991) emphasise that this time lag might be on several hundred years for long lived species, such as long-lived trees, and the result is an extinction debt (cf. Angelstam 2001, Hanski and Walsh 2004) that is difficult to assess. In some countries extinction rates due to different land use impacts are available (see Müller-Wenk 1998, Köllner 2000), but for most areas of the world this is not the situation.

The abundance of the species present is also of interest. Chapin et al. (2000) stress the importance of abundance for ecosystem functioning, and Didham et al. (1996) show that even if a species is present in an ecosystem, the ecosystem might function as if the species is absent if the abundance falls under a certain level. Hengeveld et al. (1995) emphasise that the number of species alone is not enough to evaluate diversity, but also e.g. evenness should be taken into account.

The underlying cause of these problems is the fact that biodiversity as defined by UNEP (1992) cannot be assessed directly. Several authors within the field of biodiversity have thus started to focus on indirect indicators and focus on conditions known to be important for biodiversity. Hansson (2000) states that a biodiversity indicator might as well be a

[1] The term 'condition' is used here for all environmental factors influencing the species probability to survive. Thus, it includes both what in ecological terms is recognized as conditions (abiotic environmental factors which varies in space and time, and to which organisms are differentially responsive, cf. Begon et al. 1986) and resources (all things consumed by an organism, cf. Tilman 1982).

structural component, a process, or some other feature of the biological system that ensures maintenance or restoration of the most important aspects of biodiversity when present. From this point of view Larsson (2001) focus on the key factors affecting biodiversity. For forests, this means to recognise that biodiversity is dependent on the structure of stands and landscapes, the forest formatting trees and the management and disturbance regimes they experience. Larsson (2001) identifies in total 17 key factors for assessing biodiversity in European forests. These are used as a basis for the proposed methodology in this paper.

2 Proposal of Methodology

Three different aspects must be assessed to quantify the land use impact on biodiversity (cf. Milà i Canals et al. 2007). First, a quality measure must be established and assessed. Second, the area affected must be recognized and third, the duration of the impact. This is shown schematically in **Fig. 1**. Due to changes in land use at time t_1 the quality declines from Q_0 to Q_1. At t_2 the land use stops and the area is left for relaxation, and at t_3 the quality has been restored to Q_0. The changes in quality are given by the bold line and the total impact is given by the shaded volume. Other outcomes are possible, e.g. different quality at t_0 and t_3, gradually changes in quality between t_1 and t_2 etc. (see Lindeijer et al. 2002).

2.1 The quality of an area in terms of biodiversity

In the absence of possibilities to assess biodiversity directly, it is here proposed to assess biodiversity indirectly by means of three factors:

- the Ecosystem Scarcity (*ES*)
- the Ecosystem Vulnerability (*EV*)
- the Conditions for Maintained Biodiversity (*CMB*)

Quality (*Q*) at a given location and time can be assessed as a product of these three factors:

$$Q = ES \times EV \times CMB \tag{1}$$

2.1.1 Ecosystem scarcity (*ES*)

This indicator was introduced by Weidema and Lindeijer (2001). The rationale for using *ES* as an indicator is that biodiversity linked to scarce ecosystems normally would be more vulnerable than biodiversity linked to more widespread ecosystems. The populations will in general be smaller and the extinction risk due to stochastic processes higher. Weidema and Lindeijer (2001) expressed the indicator as the inverse value of the potential area of the structure[2] (A_{pot}), resulting in the equation

$$ES = \frac{1}{A_{pot}} \tag{2}$$

This indicator can be used at different levels (biome, landscape, vegetation type etc.) depending on data availability and purpose of the study. Weidema and Lindeijer (2001) used the indicator at biome level, but data on 825 ecoregions[3] are now globally available. Since the analysis can be performed at different levels, the indicator score should be normalized following the equation

$$ES = 1 - \frac{A_{pot}}{A_{max}} \tag{3}$$

where

A_{max} is the potential area of the most widespread structure at the relevant level. The structure with the highest scarcity then gets a score close to 1, while other structures have scores relative to this. This normalization follows a linear relationship between potential area and biodiversity quality. Other relationships are of course possible but will not be discussed here.

[2] The term 'structure' is used here to indicate that this could be used at different levels; biome, landscape, ecosystem, vegetation type etc, and structure is used as a level independent term.

[3] As defined by Olson et al. (2001) including 867 ecoregions, while data on 825 is available on http://www.worldwildlife.org/wildfinder/

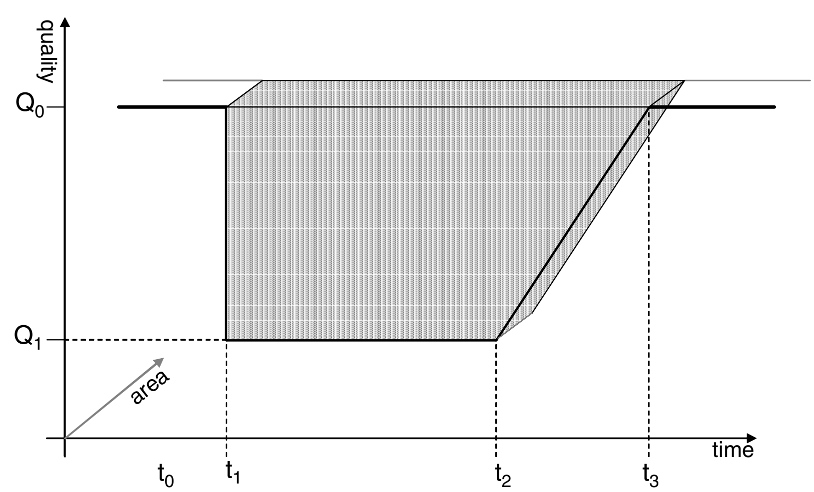

Fig. 1: Changes in land quality and total impact due to land use changes (adapted from Lindeijer et al. 2002)

To be able to use this as an indicator for forestry at different sites, it is necessary to have area factors for different forest types[4]. Comprehensive classification systems for vegetation types exists (e.g. Fremstad (1997) for Norway and Påhlsson (1998) for the Nordic countries), but at present data on potential distribution are in general absent and data on ecoregions are used.

2.1.2 Ecosystem vulnerability (*EV*)

Ecosystem Vulnerability (*EV*) is introduced as an indicator to give information about the present total area pressure to an ecosystem type and relate the existing area of an ecosystem to the potential area. The rationale is that the more of an ecosystem that is lost, the more valuable is the remaining areas. This is a consequence of the species-area relationship (MacArthur and Wilson 1967). As with the previous indicator, this can be used at different levels depending on data availability and purpose of the study.

Peter et al. (1998)[5] proposed the formula

$$EV = \frac{1}{1 - \text{fraction lost}} \tag{4}$$

while Weidema and Lindeijer (2001) proposed the formula

$$EV = \left(\frac{A_{exi}}{A_{pot}}\right)^{z-1} \tag{5}$$

A_{exi} is the existing area of the structure and A_{pot} is the potential area. z varies between different ecosystems (Hengeveld et al. 1995), but are often given the value 0.25 (MacArthur and Wilson 1967).

Both these proposed formulas give the range $[1, \infty]$ and must thus be normalized. One possibility is to normalize in the same manner as with *ES* and give the most vulnerable structure the score 1 and other structures scores relative to this.

However, data on *EV* is hard to find on an appropriate level and in most cases it will be necessary to use proxy values. Information on conservation status can be used and is often readily available. Fremstad and Moen (2001) classify Norwegian vegetation types (cf. Fremstad 1997) in the same scale as used in species red lists. World Wildlife Fund provides a three grade scale on conservation status for the ecoregions of the world[6]. In the absence of better data, this is made use of and ecoregions with the conservation status critical are given the score 1.0, ecoregions with status vulnerable are given the score 0.5 and intact ecoregions are given the score 0.1.

[4] 'Forest types' is used here for distinct formations of forests and can be defined on different scales. When ecoregions are used, only two forest types are relevant for boreal forests in Norway (PA0520 and PA0608), but if e.g. the classification in Fremstad (1997) is used, 5 major forest types are identified with in total 24 subtypes.
[5] Originally named the 'area factor' by Peter et al. (1998).
[6] See http://www.worldwildlife.org/wildfinder/ and Olson and Dinerstein (1998).

2.1.3 Conditions for maintained biodiversity (*CMB*)

The indicators on Ecosystem Scarcity and Ecosystem Vulnerability give information on the intrinsic biodiversity value of an area, while the indicator on Conditions for Maintained Biodiversity (*CMB*) gives information on the present conditions for the biodiversity in the area; is it intact, or is it reduced. Under some circumstances it might even be improved, which will be described below.

CMB is in fact an index composed by indicators known to be important for biodiversity in the particular structure. *CMB* must therefore be ecosystem specific since the key factors (cf. Larsson 2001) are different in different ecosystems. The number of key factors will also vary. Hence, it is here proposed to assess *CMB* as

$$CMB = 1 - \frac{\sum_{i=1}^{n} KF_i}{\sum_{i=1}^{n} KF_{i,max}} \tag{6}$$

where

KF_i are the different key factors identified. Larsson (2001) suggests using a four level scale for the status of the key factor:

0 – no impact
1 – slight impact
2 – moderate impact
3 – major impact

In addition, the relative importance of the different key factors for biodiversity must be determined. It is here proposed to use the scale 1 – slight importance, 2 – moderate importance, and 3 – major importance and multiply the status score with this factor. As a consequence, an indicator with slight importance has the scale [0,1,2,3] while an indicator with major importance has the scale [0, 3, 6, 9]. $KF_{i,max}$ is then the maximum score for KF_i, giving *CMB* the range [0,1] independent of the number of included key factors. A *CMB* score on 1 indicate that the biodiversity in the area is not affected, while a score on 0 indicate that the land use is devastating for the biodiversity. Larsson (2001) has proposed a range of possible key factors for European forest and an example of how this can be used is presented below.

When the framework proposed by Milà i Canals et al. (2007) is used, the quality of an area before a land use intervention (at time t_0 in Fig. 1), is given as

$$Q_{t0} = ES \times EV \times CMB_{t0} \tag{7}$$

while the quality of the same area after the land use intervention (at t_1 in Fig. 1), is given as

$$Q_{t1} = ES \times EV \times CMB_{t1} \tag{8}$$

It follows from this that if the area is undisturbed by human activities before the intervention, Q_{t0} is simply the product of *ES* and *EV*. It also follows from this that the land use impact might be positive if $CMB_{t1}>CMB_{t0}$, e.g. as a result of restoration or improved land management.

2.2 Spatial and temporal impact

The duration of the intervention in time and space must be assessed together with the quality difference. In a forest, this is defined as the time and area necessary for regrowth of the amount of timber harvested. This means that if the annual increment is 5m³/ha, 0.2 ha×y is needed to provide 1 m³ of logged wood. As a consequence, the spatial and temporal impact will be lower in forests with higher productivity.

3 Results: Land Use Impact in a Case Study of Forestry on Norway

A life cycle assessment of forestry operations in Norway is presented in Michelsen et al. (in prep.). The functional unit is 1 m³ round wood logs under bark delivered at the gate of a factory. Forestry and silviculture operations, such as seedling production, planting, soil scarification, cleaning of unwanted vegetation, logging and construction of forest roads are included. The assessment is performed in cooperation with ALLSKOG BA which represents the majority of forest owners in the western and northern parts of Norway and the analysis is valid for this area (see Michelsen et al. in prep. for details). However, only 'traditional' impact categories are included. In the present paper land use impact on biodiversity in this case is assessed following the proposed methodology.

3.1 Impacts on quality

3.1.1 Intrinsic quality score

Since the case study is of logging of spruce, the logging can take place in two different ecoregions; in region PA0608 Scandinavian and Russian taiga or in the less distributed PA0520 Scandinavian coastal conifer forests (cf. Olson and Dinerstein 1998). The distribution of PA0608 is 2,156,900 km² while the distribution of PA0520 is 19,300 km². The ecoregion with the largest distribution is PA1327 Sahara desert with 4,639,900 km². This is used as A_{max} in the calculation of *ES* (cf. Equation 3, Table 1). The conservation status for both PA0608 and PA0520 is critical. All data are from World Wildlife Fund's Wildfinder. These data are used to calculate *ES*×*EV* for the two ecoregions as shown in **Table 1**.

3.1.2 Definition and assessment of key factors

Larsson (2001) identifies in total 17 key factors for biodiversity in European forests. Not all are of equal importance for boreal forests. Similar lists are proposed by others (e.g. Schweinle 2002, Stokland et al. 2003). The main problem with most of the key factors is the scaling, i.e. how much of a particular key factor is needed for the scores from 0 (no impact) to 3 (major impact) and what is the relative importance of the key factors. As mentioned, the advantage with the proposed methodology is the possibility to start with a few key factors and subsequently prolong the list. According to Hanski and Walsh (2004) the two most important factors for decline of biodiversity in boreal forests are the reduced amount of decaying wood and loss of the most diverse forest formations. Based on this fact and combined with present data availability, three key factors are here included in a first proposal:

- amount of decaying wood
- areas set aside
- introduction of alien tree species

It is important to underline that these key factors are not independent of each other. In particular, the size of the areas set aside has consequences for the targets of the others. However, if the areas set aside should be sufficient to maintain the biodiversity within boreal forests, it would probably be necessary to set as much as 60% of the areas aside (Framstad et al. 2002). This is not realistic, so conservation of biodiversity must be based on both areas set aside and sustainable forestry (Bengtsson et al. 2000, Framstad et al. 2002). The proposed threshold values for the other key factors must hence be seen in relation with the values for areas set aside.

The species' probability to survive changes in the amount of suitable habitats might in principle follow two response curves. First, there is a linear relationship shown as response type I in **Fig. 2**. However, in most cases there is a non-linear relationship where the amount of suitable habitats has to exceed a threshold value for the species to be able to maintain or establish a viable population (Hanski and Walsh 2004). This is shown as response type II in Fig. 2. It is believed that at least threatened species follow this response curve (Hanski and Walsh 2004). It is also possible to com-

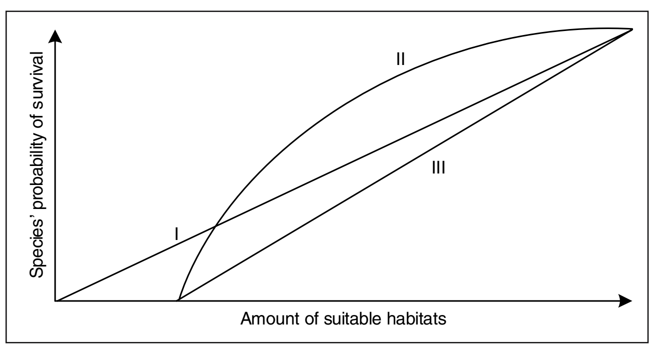

Fig. 2: Three possibilities for species' probability to survive changes in habitat quality (see text for details)

Table 1: Intrinsic quality of the two relevant ecoregions (see text for calculation details)

Ecoregion	Potential area (A_{pot})	Ecosystem Scarcity (*ES*)	Conservation status	Ecosystem Vulnerability (*EV*)	*ES*×*EV*
PA0520	19 300	0.9958	1 – critical	1.0	0.9958
PA0608	2 156 900	0.5351	1 – critical	1.0	0.5351

Table 2: Proposed scale for the key factor 'Amount of decaying wood'

Amount of decaying wood	Impact
> 20 m^3/ha	0 – no impact
10–20 m^3/ha	1 – slight impact
5–10 m^3/ha	2 – moderate impact
< 5 m^3/ha	3 – major impact

bine these two and use a linear relationship with a threshold value (response type III in Fig. 2). Identification of the most probable response curve is one element in defining the severity of the key factors at different impact levels.

Amount of decaying wood. It is well documented that present level of dead wood in managed boreal forests is far below what is found in undisturbed boreal forests, and Siitonen (2001) estimates that the decline is as high as 90–98 percent due to forestry. In Norway the average in productive forests are at time being 8.3 m^3 dead wood/ha (Hobbelstad et al. 2004).

There are different opinions on how much decaying wood that is necessary to prevent extinction of species depending on dead wood, but an estimate on 20 m^3/ha in managed forests seems to be a minimum (Hanski and Walsh 2004). Framstad et al. (2002) claim that present level of dead wood might result in an extinction of half of the organisms depending on dead wood in Norway. A first proposal of impact on this key factor is given in **Table 2**.

Areas set aside. Areas set aside are important since it is unlikely that the normal forest dynamics can be preserved within managed forests, such as forest fires, storm felling and browsing. It is also important to preserve the ecosystems capacity to evolve and function also under changed environmental conditions, e.g. climatic changes (Aarts and Nienhuis 1999). It is of course not only the total size of the area that matters; it is important to both have representative areas and large areas (e.g. Framstad et al. 2002). However, if we assume that areas are set aside as a result of a conservation plan, it is possible to assume that this is taken care of and hence only focus on total area as a key factor.

There are conflicting views on how much that is necessary to set aside, but in a combination with more sustainable forestry, there seems to be an agreement that about 10% should be sufficient (Framstad et al. 2002, Hanski and Walsh 2004). In Norway, about 2% of the areas are at present set aside. Half of this is done through establishment of national parks and nature reserves (Framstad et al. 2002), while the second half is a result of the PEFC[7]-standards used by almost all forest owners in Norway (e.g. Sverdrup-Thygeson et al. 2004). A first proposal of impact on this key factor is given in **Table 3**.

Table 3: Proposed scale for the key factor 'Area set aside'

Areas set aside	Impact
10%	0 – no impact
6–10%	1 – slight impact
1–6%	2 – moderate impact
<1%	3 – major impact

[7] Programme for the Endorsement of Forest Certification schemes, http://www.pefc.org/

Table 4: Proposed scale for the key factor 'Percentage of alien tree species cover'

Percentage of alien tree species cover	Impact
0%	0 – no impact
0–10%	1 – slight impact
10–25%	2 – moderate impact
>25%	3 – major impact

Introduction of alien tree species. Introduction of alien species are known to have a severe effect on ecosystems (eg. Clay 2003, Eppinga et al. 2006), and when the forest formatting trees are changed, the whole ecosystem is affected (Cushman et al. 1995, Larsson 2001, Stokland et al. 2003). Different tree species produce e.g. litter of different amount and quality, and provides different kinds of shelter etc. Stokland et al. (2003) distinguish between local introductions and long distance introductions. A relevant example on the first is introduction of *Picea abies* in *Betula*-stands in the western and northern parts of Norway, while examples on the second are introduction of *Picea sitchensis* and *Larix spp*. in spruce (*Picea abies*) forests.

Introduced tree species constitute 2.4% of the forests in Norway (Stokland et al. 2003). Impact values are here not well developed, but a first proposal on this key factor is given in **Table 4**.

Assessment of conditions for maintained biodiversity. The three proposed key factors are not weighted to each other, but are assumed to have the same impact given the same score. Following equation 6 the present average value for *CMB* in conifer boreal forests in Norway is thus

$$CMB = 1 - \frac{\sum_{i=1}^{n} KF_i}{\sum_{i=1}^{n} KF_{i,max}} = 1 - \frac{2+2+1}{3+3+3} = 0.44 \quad (9)$$

Several other key factors could be considered, e.g. cutting regime, tree species composition (in particular amount of deciduous trees in boreal coniferous forests), regeneration methods (area left for natural regeneration), ditching, forest road density and amount of large trees (cf. Larsson 2001, Schweinle 2002, Stokland et al. 2003). These are however not included in this first proposal.

3.2 Spatial and temporal impact

The annual increment of conifer trees in productive forests is on average 2.3 m^3/ha in Norway (Stokland et al. 2003). Thus, for the production of the functional unit of 1 m^3, 0.435 ha×y is needed.

3.3 Total impact of land use

Milà i Canals et al. (2007) propose to use the dynamic reference situation for assessing quality changes. In the case study presented in this paper, it is assumed that the forest already is altered due to centuries of forestry, and the land use in the

case study represent a postponement of the natural processes that eventually will bring the area back to its natural state and quality (=$ES \times EV$).

Further, it is assumed that the relaxation time is equal to the rotation time in the forest. The total impact caused by land use can then be assessed as shown in **Fig. 3a**. The time and area needed for one rotation period (t_{rot}) is as shown above 0.435 ha×y. The quality due to the forestry operations (assuming forestry in ecoregion PA0608) is given by

$$Q_{t1} = ES \times EV \times CMB_{t1} = 0.535 \times 1 \times 0.44 = 0.235 \quad (10)$$

This represents a postponement of a potential quality after relaxation, given by

$$Q_{trel} = ES \times EV = 0.535 \times 1 = 0.535 \quad (11)$$

The quality difference ($\Delta Q = Q_{trel} - Q_{t1}$) is thus 0.3 for a duration of 0.435 ha×y, giving a total impact of land use on biodiversity expressed as 0.131 ΔQ×ha×y.

The proposed assessment visualized in Fig. 3a might be an underestimation of the actual impact. The temporal impact is assumed to be equal to the time needed for the forest to regrow (t_{rot}), which might be an underestimation of the time needed for biodiversity to recover (cf. Duffy and Meier 1992, Müller-Wenk 1998). According to Milà i Canals et al. (2007) the dynamic reference situation should be used to assess the land use impact and the total impact from land use should therefore be calculated as 'II' in **Fig. 3b**. No attempt is made to verify if this is significant for a rather slow growing forest, but this could be of major importance, particularly in other ecosystems.

In addition, no attempt to include the transformation impact is done. The transformation impact is caused by the initial transformation of an area from an undisturbed forest to a managed forest, and the total transformation impact is shown as 'I' in Fig. 3b. However, this impact must be allocated to all timber logged in this area, and if there have been many rotations, this will with time become insignificant. It is not made any attempt to verify if this is the situation in this case.

3.4 Sensitivity of the methodology

In the previous example average values for forestry in ecoregion PA0608 are used. Two different cases are likely to occur as well. First, the logging can be situated in ecoregion PA0520. Here, a slightly higher annual increment can be assumed (Stokland et al. 2003) and the value 3.0 m³/ha is used here. The area and time needed to provide 1 m³ of wood is then 0.333 ha×y, but the intrinsic quality is significantly higher due to the higher ecosystem scarcity (cf. Table 1).

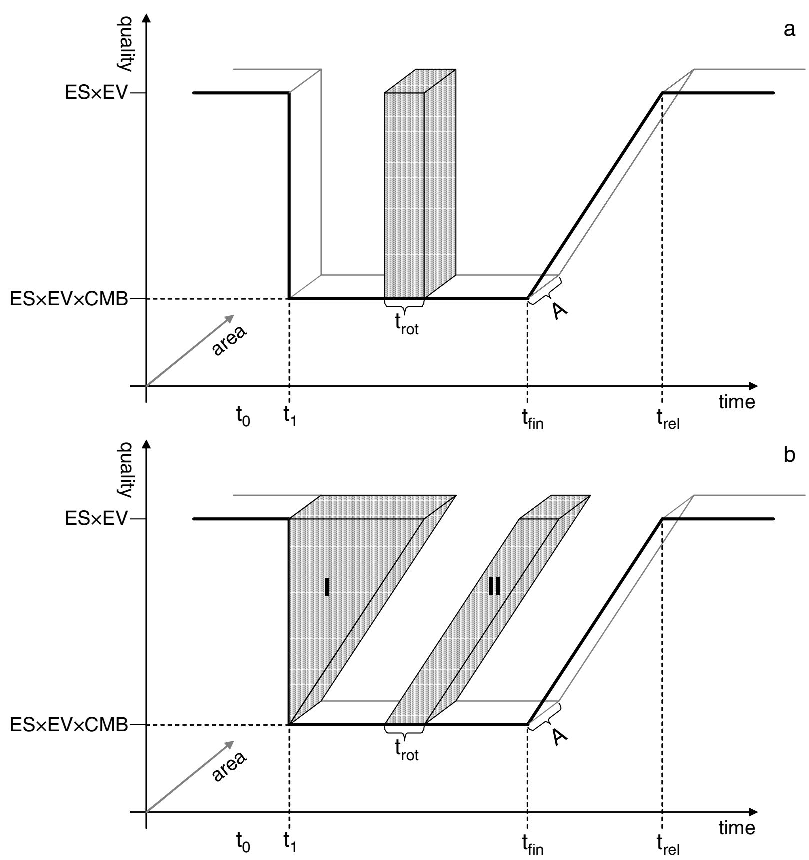

Fig. 3: A graphical interpretation of land use impact on biodiversity (see text for details)

Table 5: Differences in land use impact on biodiversity due to different ecoregions and changes in forestry regime

Case	$ES \times EV$	CMB_{t1}	ΔQ	ha×y	ΔQ×ha×y
PA0608	0.535	0.44	0.300	0.435	0.131
PA0608, 6% set aside	0.535	0.56	0.235	0.435	0.102
PA0520	0.996	0.44	0.558	0.333	0.186
PA0520, 6% set aside	0.996	0.56	0.438	0.333	0.150

It is also possible to assume that the areas set aside are increased to slightly above 6%. This can either be a result of implementing the Swedish FSC[8]-standard (The Swedish FSC Council 2000) instead of the Norwegian PEFC-standard (Living Forests 1998), intensified demands in the PEFC-standard[9], or as a result of increased areas of forest reserves following the minima recommendation of Framstad et al. (2002). The impact on this key factor might thus decline from 2 to 1 (cf. Eq. 9).

These two options can of course be combined and the results are presented in **Table 5**.

3.5 Relative importance of land use on biodiversity

It is controversial to compare different impact categories in LCA and weighting factors for comparing land use to other impact categories will not be suggested here. However, the importance of this category seems indisputable. Intuitively, this must be the situation since land use is the single most important cause for loss of biodiversity (cf. Pimm et al. 1995, Chapin et al. 1998, Müller-Wenk 1998, Chapin et al. 2000, Sala et al. 2000), which again is one of the largest environmental problems (cf. Diaz and Cabido 2001). Seppälä et al. (1998) have concluded that loss of biodiversity is the major environmental problem caused by forestry.

In Eco-indicator 99 there are proposed weighting factors that enable comparison of the impact of land use to ecosystem quality to other impact categories (Goedkoop and Spriensma 2001). As an example, acidification is given the weight 1.04 PDF ym^2/kg SO_X (see Goedkoop and Spriensma 2001). In the presented case, the total emissions are 0.113 kg SO_X (Michelsen et al. in prep), giving an impact of 0.118 PDF ym^2.

In comparison, Hanski and Walsh (2004) states that 1,000 of the 20,000 forest living species in Finland are threatened by extinction due to present forestry practice, giving a PDF of 0.05. Assuming that the number for Norwegian forestry is equivalent, this number can be multiplied with the necessary space and time needed for logging 1 m^3 of timber. The impact will then be 217.5 PDF ym^2, an impact more than 1,800 times higher than the impact due to acidification in this particular case.

4 Discussion

In this paper a new methodology for how land use impacts on biodiversity can be included in LCA is presented. The methodology is also used on a case study of logging in Norway.

The proposed methodology is shown to distinguish both between different forestry management regimes and forestry in different ecoregions (cf. Table 5). Logging in PA0520 Scandinavian coastal conifer forests represents approximately a 40% increase in the impact compared to logging in PA0608 Scandinavian and Russian taiga according to the proposed methodology. The figure makes sense (cf. Framstad et al. 2002), but the size of the difference can at present not be validated. Also, the hypothetical increase of areas set aside reduces the impact about 20%. More work is needed to verify and adjust these results, in particular on the scales of the key factors. Nevertheless, this paper represents a proposal for how such methodologies can be developed and what indicators on biodiversity should be further investigated.

Selection and scaling of key factors are critical steps. In this paper three key factors are introduced. These are assumed to be among the most important for biodiversity in boreal forests (cf. Hanski and Walsh 2004), but there are obviously others (Larsson 2001, Schweinle 2002, Stokland et al. 2003). It is probably not possible to determine the scale and mutual importance of these on a purely scientific basis with present knowledge (cf. Bennett and Adams 2004), but expert judgements are often seen as a good approximation (Seppälä et al. 1998, Scholes and Biggs 2005).

When different forestry regimes are to be compared, it must be determined what geographical range is to be used for assessing the selected key factors. As an example, within a spruce plantation on the western coast of Norway, the entire tree cover will in most cases consist of introduced *Picea abies*. However, for the landscape as a whole, the plantations constitute a rather small proportion of the area. In this paper, average values for Norway are used, but in most cases it will probably be more appropriate to set the scores according to the state within the borders of a decision-making unit. This might be a single forest owner, or as in the study described in Michelsen et al. (in prep.), within the borders of a forest owner association.

The temporal scale is also assessed in a simplified way in this paper. In Fig. 3 the solid line can be interpreted as the quality difference over the time forestry is performed in the area (cf. Milà i Canals et al. 2007). The first forestry operations take place at t_1 and forestry is carried out until t_{fin} when the area is left for relaxation. The relaxation is completed at t_{rel}. The temporal impact is here assumed to be equal to the time needed for the forest to regrow (t_{rot}), but as pointed out, this is most likely an underestimation. The significance of this simplification must be evaluated and more accurate relaxation times are needed. The potential significance of the transformation impact must also be further investigated (cf. Fig. 3b).

Another problem is how to deal with forestry using selective felling where rotation periods are hard to identify. One

[8] Forest Stewardship Council, http://www.fsc.org/
[9] The PEFC-standard is under revision and this is one of the issues debated, cf. http://www.nationen.no/naeringsliv/article2293073.ece (article in the Norwegian newspaper 'Nationen').

approach is to consider the entire forest (e.g. defined within the borders of a decision-making unit, as above) as the area affected and the temporal impact being the time until the next selective felling in the forest. The timber taken out in this period must be allocated to the selected area. The area × time would then be relatively large, but the quality difference relatively small. It is also possible to do this the other way around and start with the regrowth rate and see how much area and time are needed for regrowing the requested amount of timber and assess the quality difference only at these spots. The area × time would then be much smaller, but the quality difference would be higher. In theory these approaches should give the same result, but this still needs verification.

The proposed methodology assesses the changes in quality as given by the solid line in Fig. 3. However, the actual quality in terms of biodiversity will in most cases change more gradually, e.g. as a result of long lived species that are able to survive a period after the ecosystem conditions are changed. This is referred to as an extinction debt. This methodology does thus not assess the present quality, but the future quality following the present management regime.

The intrinsic quality assessment is sensitive to the size of the ecoregions. If, for instance, a ecoregion is split in several new ecoregions, the assessed quality of the areas within them will increase significantly. It must thus be assumed that the subdivision of ecoregions done by Olson et al. (2001) is done on a consistent basis. In the future, it might also be possible to use finer scales, e.g. the vegetation types identified by Fremstad (1997) and Påhlsson (1998). The consistency of the subdivision into different structures is obviously a critical part of the proposed methodology.

A problem that is not taken into consideration at this point is how seminatural vegetation should be treated, for instance forested pastures. If the assessment of the two factors Ecosystem Scarcity and Ecosystem Vulnerability is applied strictly as proposed, semi-natural vegetation types will be regarded without any value since their potential area without human influence by definition is zero. It is of course possible to argue that only natural occurring vegetation should be preserved and maintained, but this is not a common opinion and will undoubtedly result in extinction of a range of species adapted to these habitats through millennia. Studies indicate that even in temperate forests there are species that are adapted to forestry and hence would become threatened if forestry stopped (Decocq et al. 2004). This problem must be addressed if this methodology is to be used also for seminatural vegetation.

5 Conclusions and Recommendations

The importance of land use impact on biodiversity is of major importance and this should be included in LCIA, in particular when raw materials, such as wood, originate from land extensive activities. However, there is no agreed upon methodology for how this should be done, and a debate on the topic is crucial.

The proposed methodology provides a possibility to distinguish between land use impact from forestry both related to different forestry management regimes and forestry at different locations. The methodology is proposed within the framework provided by Milà i Canals et al. (2007), which is an outcome of the UNEP-STEAC Life Cycle Initiative. The scale of the differences must however be subject to further investigations, probably based on expert judgements.

More research is needed to see if this methodology can be transferred to other land use interventions than forestry in boreal forest. The intrinsic quality ($ES \times EV$) of all ecoregions in the world is readily available, but CMB-factors must be developed individually for all structures before a complete assessment can be performed.

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